The Role of Economics in Extended Producer Responsibility: Making Policy Choices and Setting Policy Goals

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Abstract

Extended producer responsibility (EPR) embodies the notion that producers should be made physically or financially responsible for the environmental impacts their products have at the end of product life. The EPR concept has taken hold in Europe and is garnering wide interest in the United States, where a variant known as "shared product responsibility" or "product stewardship" is usually the preferred approach. There are several policy instruments that are consistent with EPR—product take-back mandates, advance disposal fees, deposit-refunds, recycled content standards, and more. The EPR concept itself, however, provides little guidance about which of these instruments might be appropriate under particular conditions and for particular products. Moreover, while the EPR goal is usually focused on end-of-life environmental impacts, in the United States, at least, the goal seems to have widened to include environmental impacts throughout the product life-cycle. And even a focus on end-of-life impacts leaves the question of whether EPR is intended to deal with waste volumes, the toxic constituents of waste, the method of waste disposal, or a combination of these things. In this paper, I address three main topics: appropriate goals for EPR; conditions under which EPR should be preferred over alternative non-EPR policy instruments; and specific policy instruments that are both cost-effective and consistent with EPR principles. In the discussion of the second and third topics, I focus on the issue of "design for environment." I develop four policy "maxims" that should guide EPR policymaking. I then apply those maxims to a brief case study of electronic and electrical equipment waste.

Key Words: EPR, recycling, design for environment

JEL Classification Numbers: Q2, H2

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1. Introduction

Extended producer responsibility (EPR) embodies the notion that producers should be made physically or financially responsible for the environmental impacts their products have at the end of product life. The EPR movement began in Europe and has spread to other countries. The original impetus for it was twofold: to relieve municipalities of some of the financial burden of waste management, and to provide incentives to producers to reduce resources, use more secondary materials, and undertake product design changes to reduce waste (OECD, 2001).

Several countries set recycling rate targets and mandate take-back of a number of key products, including packaging, vehicles, electronic and electrical equipment, and batteries. For European countries, these policies are backed by European Union directives on packaging and vehicles and an expected electronics waste directive in early 2003.[1] Several state governments in the United States are considering legislation directed at electronics. In addition, a number of industry and multi-stakeholder voluntary efforts have arisen in the United States in recent years.

There are several policy instruments that are consistent with EPR—product take-back mandates, advance disposal fees, deposit-refunds, recycled content standards, and more. The EPR concept itself, however, provides little guidance about which of these instruments might be appropriate under particular conditions and for particular products. Moreover, while the EPR goal is usually focused on end-of-life environmental impacts, in the United States, at least, the

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[1] See Palmer and Walls (2002) for an overview of EPR programs in several countries, as well as a discussion of the mostly voluntary "product stewardship" efforts in the United States.

goal seems to have widened to include environmental impacts throughout the product life-cycle.[2] And even a focus on end-of-life impacts leaves the question of whether EPR is intended to deal with waste volumes, the toxic constituents of waste, the method of waste disposal, or a combination of these things. Clarifying the EPR goal is critical for choosing the right policy instrument and evaluating the success or failure of that instrument.

In this paper, I address three main topics: the appropriate goals for EPR; the conditions under which EPR should be preferred over alternative non-EPR policy instruments; and the specific policy instruments that are both cost-effective and consistent with EPR principles. In the discussion of the second and third topics, I focus on the issue of "design for environment" (DfE). DfE is one of the primary motivations for EPR and is also a major component of cost-effectiveness. At the same time, identifying feasible and low-cost policies that motivate DfE is difficult.

In addressing these three topics, I lay out four EPR policy maxims. These maxims, while by no means the only tool a policymaker needs, should provide some guidance in evaluating EPR policy alternatives.

The paper proceeds as follows. Section 2 discusses EPR policy goals, including efficiency, cost-effectiveness, and the importance of clarity in defining goals. Section 3 addresses the question of why EPR-based instruments should be used in particular situations, providing evidence from the economics literature on optimal solid waste policy instruments. Section 4 then continues the discussion of policy instruments in more detail, with a particular eye toward DfE. What feasible instruments might spur DfE and how costly will it be, in general, to design, implement, and enforce policies that have significant DfE effects? A case study of electronic and electrical equipment waste and how the four policy maxims apply to such waste is the focus of Section 5. Section 6 provides some brief concluding remarks.

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[2] See www.epa.gov/epr/about/index.html for the U.S. Environmental Protection Agency's definition of product stewardship, which clearly states that the concept covers the "environmental impacts of products."

2. The Goals of EPR

In this section of the paper, I discuss the goals of environmental policy generally and EPR policy specifically. The focus is on economically efficient outcomes. I also emphasize the importance of clarity in specifying policy goals. The economics paradigm thus provides the first maxim for EPR policy.

> *Maxim 1. The goal of EPR policy should be no different than the goal of any other environmental policy: to maximize social welfare.*

2.1. Externalities and social welfare

Production and consumption can often cause negative externalities—unintended side effects that private market transactions fail to take into account. Pollution and waste disposal are typical examples. When these problems occur, government intervention into private markets is usually called for since internalizing the externality will improve overall social welfare. The objective of environmental policy, then, should be to attain the socially optimal level of the waste or pollution—i.e., that level where the marginal social benefit of reducing the waste or pollution by one more unit is just equal to the marginal social cost of reducing it. Overall social welfare is maximized in this case.

If the target level of waste or pollution is not set at the optimal level, the policy instrument should still be chosen so as to achieve the target at least cost. Thus, the objective for environmental policy generally, as well as for EPR policy specifically, should be to attain an efficient level of the environmental externality in question. When the efficient level of the externality is unattainable for political or other reasons, the objective of EPR policy should be to achieve the desired target, whatever it may be, at least cost.

2.1.1. Polluter pays principle and EPR

Other policy concepts or ideals may be consistent with this efficiency, or cost-effectiveness, objective, but they generally do not provide enough guidance on how to choose among alternative instruments.[3] The "polluter pays principle" (PPP), for example, is often consistent with efficiency and cost-effectiveness ideals—i.e., an efficient policy instrument will be one in which the polluter pays. However, several policy instruments might be consistent with the PPP but only one of them economically efficient. Likewise, the principle of EPR—that producers should bear some physical or financial responsibility for their products at end-of-life—can be met with many different policies but only a subset of those policies will be efficient.

Among the set of policies consistent with EPR principles are: take-back mandates, recycled content standards, advance disposal fees (sometimes called advance recycling fees), and deposit-refund programs. Take-back mandates can have "opt out" provisions, allowing individual producers to meet their obligations by joining a "producer responsibility organization" (PRO) that handles all collection and recycling. The PRO, in turn, must then choose how to set the fees it charges its member firms. Alternatively, a take-back policy can require producers to take individual responsibility for their products, rather than delegating that responsibility to a PRO. Although all of these policies may be consistent with EPR and may even achieve the same waste reduction outcomes, they do not all impose the same costs on society. The goal of EPR policy should be to choose the most efficient, or cost-effective, instrument.

2.1.2. Policy objectives and policy instruments

The first step in designing and implementing an efficient or cost-effective policy is clarifying the environmental objective of the policy. This brings me to the second EPR policy maxim.

> Maxim 2. *When designing EPR policy and programs, policymakers need to precisely specify the environmental objective they are trying to achieve.*

[3] It should be pointed out that, in theoretical models, more than one specific policy instrument often can be shown to be efficient. Several studies of solid waste policy have identified multiple combinations of instruments that are efficient. Fullerton (1997) and Schöb (1997) point out that there are, in fact, an infinite number of different normalizations of the Pigovian tax rate that can all achieve the social optimum. In practice, however, when feasibility concerns are factored in, and administrative and enforcement costs are included in cost calculations, the set of efficient instruments is greatly reduced.

Among the many goals often given for EPR are: waste diversion, reduced environmental impacts from production, such as air and water pollution, less use of virgin materials in production, and reduced toxicity of products. However, one policy instrument cannot achieve all of these goals. A long-standing result in economics, in fact, is that at least as many policy instruments are needed as policy objectives (Tinbergen, 1967). Walls and Palmer (2001), in a theoretical model with life-cycle environmental impacts—specifically, air or water pollution and a solid waste byproduct during production and solid waste as a result of consumption—show that a single instrument cannot solve all problems. Thus, attempting to internalize multiple life-cycle externalities with, say, an advance disposal fee, as some have suggested (Ackerman, 1993), would be a policy mistake.

This point is particularly important in light of concerns over the use of hazardous materials in some products. In debates over how best to set policy to address post-consumer waste from electronics, for example, arguments are often made that policy instruments need to simultaneously address the volume of such waste as well as the constituents of the waste. In the latter category are usually concerns over lead and mercury. While hazardous materials and waste volumes are both legitimate environmental concerns, it is highly unlikely that they can adequately be addressed with a single policy instrument.

2.1.3. Integrated product policy

On the other hand, taking multiple problems into account when setting multiple instruments is important. Walls and Palmer (2001) obtain this result in their model. They show that even though Pigovian taxes are optimal, just as they are in a setting without life-cycle environmental impacts, the rate for each of the taxes reflects the socially optimal level of all other pollution and waste. In other words, when setting the taxes, the policymaker needs to do so simultaneously to achieve the overall social optimum. If Pigovian taxes are infeasible, as they are likely to be in many settings, an integrated policy approach is even more important. The authors show that the alternative set of efficient policy instruments includes input taxes that are set to incorporate marginal damages from multiple environmental problems. One of the most difficult tasks for policymakers is managing this simultaneity problem when setting policies. In the United States, where environmental regulations have traditionally been divided by media—e.g., air, water, and land disposal—this task is especially difficult.

2.1.4. Reducing waste

The primary objective of EPR policy should be reducing the volume of solid waste disposal. Reducing upstream externalities associated with resource extraction or with production may be important ancillary benefits and should certainly be part of the overall policy picture, but they are not the primary policy objective of EPR. Promoting recycling or recycled content of products are not worthwhile goals in and of themselves, but are worthwhile as a means of cost-effectively reducing waste disposal (Macauley and Walls, 2000). Likewise, encouraging producers to design for environment is a means to an end and not an end itself. When evaluating the degree of success of EPR policies, the primary focus should be whether they cost-effectively reduce the volume of waste.

3. Why EPR?

With this waste reduction goal in mind, an important question is when is an EPR-based policy instrument the appropriate choice for policymakers.

> *Maxim 3. EPR policies are preferred over non-EPR policies for one of two reasons: (1) illegal disposal problems or (2) poorly functioning recycling markets.*

I discuss each of these motivations for EPR in turn, beginning with a discussion of the problems presented by illegal disposal, and then turning to a second potential market failure, transaction costs in recycling markets.

3.1. Illegal disposal

3.1.1. Policy results from economics

Several theoretical studies by economists solve for optimal policy instruments when there is the possibility of illegal disposal (Dinan, 1993; Fullerton and Kinnaman, 1995; Palmer and Walls, 1997). These studies find that there are multiple combinations of instruments that can achieve the social optimum, but all recommend a combination of a product tax and recycling subsidy—usually referred to as a deposit-refund—as the instrument of choice. Fullerton and Kinnaman (1995), in the only study to explicitly incorporate the externality from illegal dumping in a general equilibrium framework, show that not only is a Pigovian tax on (legal) disposal inappropriate in this setting, but a disposal subsidy is called for in combination with the deposit-refund. The deposit-refund efficiently reduces waste disposal by combining the "output reduction effect" and the "input substitution effect" that are inherent in the Pigovian tax without that tax's attendant illegal dumping problem.[4] In other words, the Pigovian tax reduces waste by combining source reduction incentives and recycling incentives in a single instrument; the deposit-refund provides the same incentives in two separate instruments. Thus, several theoretical studies by economists show support for at least one kind of EPR policy over a non-EPR disposal fee.

3.1.2. "Bottle bill" deposit-refunds

No distinction is made in the Fullerton and Kinnaman model, nor in the other theoretical models cited above, between their conceptual deposit-refund policy instrument and the real-world "bottle bill" deposit-refund that is used in several American states. Bottle bill deposit-refunds are collected (and refunded) downstream; typically, a fixed fee—the deposit—is charged to a consumer for purchase of a container of a given size, and that fee is given back to the consumer when the container is returned. Usually the bottler, or distributor, collects the deposits from consumers via retailers; retailers pay out the refunds and typically are required to take back containers of any brands that they sell. The distributor then accepts the containers for recycling and reimburses the retailers. Who keeps the unclaimed deposits varies across states. Bottle bills usually come with high administrative and transaction costs because of the sorting requirements. Moreover, because the fees are fixed and do not vary on a weight basis, they provide little incentive for source reduction. [5]

The deposit-refunds that economists advocate are more general than the bottle bill prototype. They are simply a combination of a product tax and a recycling subsidy, usually set on the basis of product weight.[6] The tax is passed on to consumers in the form of higher product

[4] See Spulber (1985) and Fullerton and Wolverton (1999) for more on these two effects embodied in Pigovian taxes.

[5] California has a different—and, by most accounts, more efficient—system in which the state operates redemption centers and arranges for recycling, with no sorting by brands and no involvement by distributors on the back end. The unclaimed deposits in California go to the state, which then distributes them as grants to nonprofit recycling centers and to municipal governments for the containers they collect (typically, through curbside recycling programs). For more on U.S. bottle bills, see www.bottlebill.org.

[6] Fullerton and Wolverton (1999) espouse the idea for more than just solid waste problems and refer to the tax/subsidy combination as a "two-part instrument" (2PI).

prices, but the producer may be the one who actually makes the tax payment to the government. Similarly, the recycling subsidy should make itself felt by consumers but could be paid by the government to collectors and/or processors of secondary materials.

3.1.3. The UCTS

Palmer and Walls (1999) refer to a tax/subsidy combination in which producers of intermediate goods pay the per-pound tax and collectors of used products receive the per-pound subsidy as an "upstream combined tax/subsidy," or UCTS. Palmer et al. (1997) provide some quantitative evidence on the cost savings from the UCTS over a product tax alone—i.e., an advance disposal fee (ADF)—or a recycling subsidy alone, which is not an EPR-based policy instrument. They parameterize a simple model of waste disposal using estimated demand and supply elasticities and 1990 U.S. prices and quantities for five materials commonly in the municipal solid waste stream—aluminum, steel, glass, paper, and plastic. They find that, for any given percentage reduction in waste disposal, the UCTS is the least costly option, followed by the ADF; the recycling subsidy is the most costly approach. For a 10% reduction in waste from 1990 levels, they calculate that it would be necessary to implement an across-the-board UCTS of $45, an ADF of $85, or a recycling subsidy of $98. Thus, EPR-based policies—the UCTS or an ADF, in this case—are less costly ways to achieve waste-reduction goals than a non-EPR policy, a recycling subsidy.

The UCTS is the most cost-effective of the three because it encourages *both* recycling and source reduction. The ADF, on the other hand, encourages only source reduction, and the recycling subsidy only recycling. Thus, the latter two instruments have to work harder to achieve the same reduction in waste that the UCTS achieves. This proves to be particularly difficult for a recycling subsidy because the cost of recycling some products—plastics, in particular—is quite high. The recycling subsidy result would hold for any policy instrument that targets only recycling—recycling rate mandates, recycled content standards, and investment tax credits for recycling, to name just three. Because these instruments provide no incentive to reduce output or reduce the weight of products, they are unnecessarily costly ways to reduce waste disposal.

These quantitative findings by Palmer et al. thus provide more support for EPR policies as opposed to policies that target downstream recycling markets. I must emphasize again, however, that cost-effectiveness ultimately should be the deciding factor in choosing policy instruments. Both the ADF and the UCTS can be characterized as EPR policies, but the UCTS is the preferred instrument because it achieves any given reduction in waste disposal at the lowest possible cost.

3.2. Transaction costs in recycling markets and DfE

In the Palmer et al. empirical study and all of the theoretical studies cited above, the issue of product design and its effects on recycling are ignored. Moreover, the recycling market itself is not explicitly incorporated in the models. The focus of these studies is to identify optimal policies in cases where illegal disposal precludes the use of a disposal fee. But even in cases where disposal fees are feasible, there can be a rationale for EPR.[7] This rationale comes from the existence of imperfectly functioning recycling markets.

3.2.1. Modeling transaction costs in recycling markets

Calcott and Walls (2000, 2001) develop a theoretical general equilibrium model in which heterogeneous producers make design choices that affect the recyclability of their products. The authors explicitly incorporate the recycling sector in their model: private profit-maximizing recyclers collect used products from consumers, process them, and resell to producers; the more recyclable a given product is, the lower the processing costs.[8] The authors find that if recycling markets work perfectly—that is, if recyclers pay consumers for used products and the prices they pay vary with the degree of product recyclability—then a Pigovian tax on disposal can yield a first-best, efficient level of waste disposal, recycling, and design for environment. If, however, recycling markets do not work perfectly—more specifically, if it is too difficult and costly for recyclers to pay prices that vary with the degree of the products' recyclability—then a first-best outcome cannot be reached. Calcott and Walls argue that in reality, recycling markets probably do not work perfectly. It is costly to collect and transport recyclables, and it is difficult for

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[7] Unit-based pricing (UBP) of residential waste is becoming quite common in the United States and elsewhere, and with, so far, no documented serious illegal disposal outcomes. See Miranda et al. (1998) for a useful survey of U.S. programs. Most U.S. programs set prices by the container (either bags or cans of a particular size). Linderhof et al. (2001) report on weight-based pricing used in a Dutch community. According to Allers (2002), a co-author of that study, illegal dumping was monitored carefully by the town and was found to be "minimal, but not nonexistent."

[8] Two other studies that address DfE concerns are Fullerton and Wu (1998) and Eichner and Pethig (2001). Producers are homogeneous in the model constructed by Fullerton and Wu. In addition, they provide no explicit treatment of the role of recyclers or of recycling costs. Eichner and Pethig (2001) do model recycling costs, but treat recyclability as a proportion of a product's material content that is of a particular type. The Calcott and Walls definition of recyclability is more general.

recyclers to sort products according to their recyclability and pay consumers a price based on that degree of recyclability.

3.2.2. The second-best outcome

Calcott and Walls (2001) find that although transaction costs in recycling markets preclude achieving the social optimum, a constrained, second-best optimum can be reached. A product tax/recycling subsidy (i.e., a UCTS) combined with a disposal tax set at less than the Pigovian rate—that is, less than the full marginal social costs of disposal—will achieve the second-best outcome.[9]

In the Calcott and Walls models, the constrained optimum is the best outcome that can be achieved given the transaction costs in recycling markets. It is important to emphasize this point. If firms have a choice over the design of their products, in particular over the recyclability of their products, and recycling markets fail to work perfectly, then price signals are not transmitted from consumers and recyclers back upstream to producers. It is extremely difficult to design a *feasible* policy that overcomes this problem.[10]

On the other hand, the Calcott and Walls results are encouraging: the second-best outcome is attainable with a simple set of policy instruments, a set that falls under the EPR umbrella. The tax and subsidy give producers the incentive to make their products recyclable enough to get over the threshold at which profit-maximizing recyclers (who receive a subsidy from the government) will accept the products. Above that threshold, the existence of working recycling markets—even imperfect ones—spur DfE. So although the social optimum is not attained, this second-best outcome is one in which there is less consumption, less waste, more recycling, and a higher degree of recyclability of products than in the free market.

[9] Calcott and Walls also find an alternative policy that yields the constrained optimum: a product tax/recycling subsidy in which the product tax takes on one of two rates depending on whether the product is recyclable enough to be accepted by processors—that is, processors do not incur a loss if they recycle it. The tax on products that do not reach that recyclability threshold is the standard Pigovian tax and thus can be viewed as an advance disposal fee. Products that meet the threshold receive a subsidy when they are recycled that is equal to the tax paid up-front. No disposal tax is necessary with this second set of instruments.

4. EPR Policy and DfE

The combination of a UCTS and a modest disposal fee, as advocated by Calcott and Walls, along with the existence of recycling markets—even those that work imperfectly—provide some incentives for DfE. Most environmentalists and environmental policymakers, however, would probably view the effects as insufficient. Also, on the face of it, a uniform product tax and recycling subsidy may seem to many observers to be incapable of having any real impact on product design.

There could be merit to these arguments. After all, some faith in markets—even imperfectly functioning ones—is required to believe that the Calcott and Walls policy motivates any DfE. In their model, for some products at some level of secondary material prices, the value of the materials being recycled outweighs the transaction costs in recycling markets, and this is what brings about product design changes to increase recyclability. If those signals don't work at all for some products, then the UCTS and disposal fee might be inadequate.

On the other hand, designing, implementing, and enforcing a policy that does have significant effects on product design is likely to be difficult and costly. This brings us to our fourth EPR policy maxim.

> *Maxim 4*. *While DfE provides the primary motivation for EPR, designing feasible and low-cost policies that promote DfE is inherently difficult.*

We present two economic incentive-based policy instruments here that may have the potential to spur DfE: tradable recycling credits and a DfE reward system. We then return to the issue of the cost of policy implementation.

[10] Calcott and Walls (2001) show that a tax and subsidy that vary with the degree of recyclability of products can yield the first-best social optimum, but it is difficult to conceive of a situation in which policymakers would have enough information to set these instruments and have the political wherewithal to set tax rates that vary across firms.

4.1. Tradable recycling credits

The studies by economists that I reference here do not look beyond tax and subsidy-based policy instruments. In particular, the studies do not look at other EPR policies such as take-back and a policy that is perhaps most worthy of further study, tradable recycling credits.[11] Tradable recycling credits are similar in spirit to tradable emissions permits (Tietenberg, 1985). One important difference between the two approaches is that a tradable emissions permit system typically is associated with a cap on total emissions from all sources, with trading allowed between sources, whereas a tradable recycling credit system imposes a minimum recycling level or rate on a particular industry and allows trading between responsible parties to reduce the cost of achieving that minimum level.[12]

A tradable recycling credit program for electronics, for example, might work as follows. Every manufacturer or importer would be required to meet a recycling rate target for its products. The target could be an overall weight target, such as 50% of the weight of the product must be recycled, or a set of specific targets by component material type. Producers could do the recycling themselves, or they could pay a recycler to do it or—and this is the interesting twist on current policies in Europe and elsewhere—they could purchase credits from others who have recycled more than their own obligation. Recyclers would be required to keep track of what they recycled by brand. At the end of the year, producers would have to show that they had met the recycling target or hold enough credits purchased from others to comply with the target.[13]

The virtue of a tradable credit system is the flexibility it has over a system in which each firm must recycle a certain percentage of its products. Firms whose products are particularly

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[11] Fullerton and Wu (1998) look at a variety of instruments and combinations of instruments. They purport to include take-back, but it is modeled by simply having producers pay disposal costs, and who pays has no efficiency implications. Moreover, producers in the Fullerton and Wu model are homogeneous, so standard-based instruments that lack flexibility across producers look as efficient as price-based ones.

[12] Another analogy would be a renewable energy portfolio standard in which electricity producers are required either to produce a minimum portion of their electricity using renewable energy sources such as wind or solar or to hold credits showing that another generator has produced the requisite amount of renewable energy. For more information, see Clemmer et al. (1999)

[13] This approach is different from the approach described in a report for the European Commission, *Tradable Certificates for Recycling of Waste Electrical and Electronic Equipment,* produced by Environmental Resources Management (1999). Under that scheme, credits are awarded to the company that pays for the recycling of the electronics equipment and not to the company that originally produced or imported it. In such a system, recyclers would not keep track of exactly which firms' products they are recycling. The costs of such a scheme might be lower than the one we describe, but so are the incentives for DfE.

difficult to recycle may choose to purchase credits, whereas firms whose products are recycled more easily will sell credits. Because selling credits earns firms money, the scheme should encourage firms to design products to be more recyclable. The virtue of the system over the current "take-back with PRO" model is the incentive it provides for DfE. Because the costs of recycling an individual product are borne directly by the producer of that product, the producer has the incentive to redesign the product to bring those costs down. In other words, since the costs and benefits of the design change are borne by the same entity—the producer—the policy avoids the free rider problem inherent in PRO schemes.

Of course, designing and implementing a tradable recycling credit system raises several issues and challenges. These include questions about how collection might work and the effect of different collection schemes on incentives for DfE; how to do the initial allocation of credits; whether to allow trading only within or across product types; whether to set overall recycling goals or material-specific ones; and how to set the recycling rate targets and the trading rules to deal with long-lived products. In the only existing tradable recycling credit scheme, the U.K. Packaging Waste Recovery Notes program, an individual firm's packaging is not tracked through the system (indeed, tracking is likely to be impracticable for packaging). The credit prices vary by material and are weight-based, so there could be some incentives for material substitution and dematerialization, but other DfE incentives are likely to be minimal.[14]

Perhaps most importantly from an efficiency standpoint, a tradable recycling credit system targets recycling and not source reduction, and thus falls prey to the criticism of the other recycling-focused instruments we discussed above. An optimal tradable recycling credit policy would be likely to include an ADF along with it.[15]

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[14] See Salmons (2002) for more detail and analysis of the U.K. system. Further study of the workings of the Packaging Waste Recovery Notes markets and the overall strengths and weaknesses of the scheme would provide useful information for the design of tradable recycling credit systems applied to other products in the future.

[15] This result has not been shown in the literature, to our knowledge; however, Palmer and Walls (1997) find that another instrument focused just on recycling and not source reduction, a recycled content standard, can achieve a socially optimal amount of waste disposal only if combined with an ADF (and input taxes). A recycled content standard has other problems, however, that a tradable recycling credit system might be able to avoid. The most important one is its lack of flexibility across producers.

4.2. DfE rewards

Another possibility worth exploring might be a combination of a UCTS with a system of financial rewards for the attainment of particular design objectives. The reward could come in the form of a reduced upstream product tax if particular product characteristics are met or particular recycling outcomes are reached. Alternatively, it could simply come as an end-of-year payment from the government or a private standard-setting body. It would be essential to have the reward linked directly to efforts on the part of the individual producer and not industry-wide effects. Moreover, real change would only come from financial rewards rather than positive publicity or some other kind of nonpecuniary benefit. Thus, programs such as the U.S. EPA's WasteWise Partner of the Year program or the industry-based Electronic Industries Alliance Environmental Progress Award, which involve only publicity, certificates, and the like, are not the kinds of programs that will generate substantive improvements.[16]

A DfE reward system would still mean that the government or some third party would need to determine the design objectives for purposes of rewards—a difficult task, but perhaps feasible for some products. Also, it would still be necessary to keep track of individual firms' products through the product life-cycle.

4.3. The costs of implementing policy

This second issue brings us back to the fourth EPR policy maxim: virtually any system with strong incentives for DfE could be very costly to implement. In my opinion, policymakers need to recognize and grapple with this problem sooner rather than later. There are critical trade-offs that need to be considered when deciding on policy options: simplicity and flexibility coupled with minimal incentives for DfE on the one hand, versus complexity and high administrative and monitoring costs combined with sharp DfE incentives on the other. The next section of the paper provides more discussion of these points in the context of one specific class of products in the waste stream, electronic and electrical equipment waste.

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[16] For more on the WasteWise award program, see http://www.epa.gov/wastewise/about/winners.htm; for more on EIA's environmental award, see http://www.eia.org/policy/awards.phtml.

5. E-Waste, DfE, and the Design of Policy

In this last section of the paper, I focus on a particular component of the municipal solid waste stream that is currently receiving a great deal of attention, electronics and electrical equipment waste, or so-called e-waste. E-waste provides an excellent illustration of each of the four maxims introduced above: (1) policy should be efficient or cost-effective; (2) objectives should be clarified and multiple objectives addressed with multiple policy instruments; (3) EPR is necessary to spur DfE because of poorly functioning recycling markets; and (4) effective policy is inherently difficult and costly to design, implement, and enforce. We address each of these points in turn, but first we provide some background on policy and proposed policy directed at waste electronics.

5.1. Current e-waste policies and proposed policies

In 2000, the European Union introduced its waste electronics and electrical equipment (WEEE) directive; final approval of the directive is expected in Spring 2003. The directive mandates that member countries have systems for take-back and recycling of electronic waste and sets an e-waste collection target of four kilograms per household per year by 2006. The directive also sets recycling rate targets for specific types of waste. These targets range from 50% to 75%.[17]

As with all EPR policies, the United States lags behind Europe. However, several states have introduced bills in recent years to address e-waste, and a bill was introduced into the U.S. Congress in July 2002 that would set a fee of $10 on the sale of each new computer and computer monitor in the United States. The revenues, which would be managed by EPA, would be used to fund organizations collecting, processing, reusing, or reselling used computers, monitors, and other designated devices.

California, New York, North Carolina, Nebraska, Massachusetts, Minnesota, and Georgia have all introduced bills in the past year or two to address a growing concern with end-of-life computers. The Massachusetts bill introduced in late 2001 (House bill 4716), would require any manufacturer selling a product with a cathode ray tube (CRT) to provide a system for its take-

[17] The lower target holds for small appliances such as irons and toasters; for consumer equipment such as TVs, video recorders, and radios; and for electric tools and toys. The higher target applies to large household appliances such as washers, dryers, refrigerators, and microwaves.

back and recycling. Several bills were introduced into both houses in California. In September 2002, California Senate bill 1523, which would have imposed a $10 fee on all computer monitors and televisions to fund recycling programs, was passed by both houses. Governor Gray Davis did not sign the bill, but released a statement supporting the product stewardship concept and challenging industry to set up a voluntary program.

The National Electronics Product Stewardship Initiative (NEPSI), a group of state government, industry, nongovernmental organizations, and EPA stakeholders, is trying to reach consensus on a national voluntary product stewardship agreement for electronics. The group is expected to reach some agreement, the exact form of which has not been determined, by early 2003. In present discussions, the NEPSI stakeholders are focusing their discussions on a front-end product fee to pay for recycling.[18]

5.2. Maxims 1 and 4: cost-effectiveness and the importance of DfE

By all accounts, collecting, transporting, and processing electronics is quite costly. In a 1999 Minnesota program operated by the Minnesota Office of Environmental Assistance, along with electronics companies Sony and Panasonic, Waste Management's Asset Recovery Group, and the American Plastics Council, the cost of collecting, transporting, processing, and marketing materials from the electronic equipment collected in the program averaged $448 per ton. Collection and transport together accounted for approximately 75% of this cost (Hainault, 2001; Hainault et al., 2001). A U.S. EPA (1998) study of electronic collection efforts in five communities found that costs ranged from $200 to $1,000 per ton. The Northeast Recycling Council (2002), in a national survey of municipal electronics collection programs in the United States in 2001, found that costs averaged $374 per ton across all locations and all types of programs.[19] These costs far outweigh the costs of collecting, transporting, and recycling the materials traditionally collected in municipal curbside programs.

Recycling fees paid by communities to electronics reprocessors have been reported in several studies. These fees provide further evidence on costs. The NERC study finds that fees average $330 per ton across the communities in their sample. In Massachusetts, where the state has entered into contracts with two processors to accept materials from all Massachusetts

[18] For more information about NEPSI and a list of stakeholders, see www.nepsi.org.

[19] Nearly 500 communities were sent surveys; 176 completed surveys were returned and used in the sample.

communities at a fixed price, those communities pay $260 per ton or $300 per ton (depending on volumes) and the community pays for transportation to the processor. If the processor collects from the municipality, the costs are higher. Hennepin County, Minnesota, which has one of the longest-running programs and accepts the broadest range of materials, pays $900 per ton in recycling fees (Northeast Recycling Council, 2002).

The American Plastics Council (2000) reports that the average cost to use advanced recycling methods to process plastics from electronics ranges from $460 per ton to $1,040 per ton. By most accounts, plastics are the most difficult material in e-waste, with the possible exception of CRT glass. The California Integrated Waste Management Board (2001) sponsored a survey of electronics waste processors in California in which processors were asked their costs for processing computer processing units (CPUs), televisions, and computer monitors. The reported costs—which do not include collection costs—ranged from $613 per ton for a CPU to $963 for a monitor, and as high as $1,488 for a television.[20]

There are several reasons for the high costs of managing electronics. Collection and transport costs are high because the items are bulky and often need to be transported long distances because of the relatively small number of e-waste processors. Processing costs are high for a number of reasons. Waste electronics contain a variety of materials, including steel, aluminum, copper, glass, plastic, precious metals (including gold, palladium, silver, and platinum), and other miscellaneous materials such as rubber and wood. This means that the separation, sorting, and dismantling cost component of overall reprocessing costs is large. And no single material is valuable enough and present in a large enough quantity to justify these costs. Results from collection programs show that nearly half of the material recovered from electronic equipment is metal, but only half of that metal is the type for which there is a well-developed recycling infrastructure. Plastics comprise the next highest percentage, at 33%. But an American Plastics Council (2000) study found that only 35% of the plastics collected in the Hennepin County, Minnesota, program were suitable for reprocessing because of the presence of contaminants such as paint, labels, coatings, and lamination. In the acceptable sample, nine

[20] The survey did not tell respondents exactly what costs to include but left that up to the individual respondent. The report states that costs are likely to include "hauling, processing, storage, and labor (p. 8)." The report publishes costs in 2006 dollars (because it is focusing on future cost and capacity issues) and assumes a 2.7% annual inflation rate; the numbers I report here are deflated back to 2001 dollars.

different plastic resins could be identified. Most of the resins are ones for which well-established and high-value secondary markets are lacking.[21]

These findings about the costs of electronics collection and recycling programs suggest that the benefits of such programs need to be quite high to cover current costs. Some recent research suggests that the benefits of recycling CRTs fall far short of the costs. Macauley et al. (2001) look at several policy options to reduce disposal of CRTs and find that these options would all increase disposal costs by far more than any benefits gained.[22]

The Macauley et al. findings might lead some observers to argue that, on economic efficiency grounds, there should be no government policy directed at reducing disposal of electronics waste. I do not argue this point here, except to draw attention to the fact that there seems to be growing public concern about e-waste and that concern is generating a great deal of attention on several policy fronts. What I would argue instead is that these results highlight the importance of finding cost-effective policy instruments. And cost-effective policy instruments will be those that spur product design changes that bring down the net cost of recycling. In other words, the high recycling costs indicate the significance of maxim 3 and the role DfE plays in cost-effective EPR policy.

There seem to be several targets for design changes that would reduce the net costs of recycling electronics. (1) The presence of contaminants leads to rejection of materials, thus reducing the use of labels, laminates, paints, and coatings on the plastics in electronics, as well as reducing the use of metal used with and on plastics, will improve the chances that the plastics in electronics can be recycled. (2) Reducing the different types of plastics and other materials that are used in a given product, and labeling the types that are used, will improve recyclability. (3)

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[21] The processor used in that study, MBA Polymers, experimented and had some success with producing a 100% pure stream of high impact polystyrene (HIPS) from television plastics, the most common resin in the waste sample. A more recent study using data from Hennepin County and co-sponsored by the American Plastics Council showed some improved results. Fisher et al. (2000) report that MBA accepted 100% of the sample it received as opposed to the earlier 35%; this was because better sorting and culling was done before the materials were shipped. Moreover, MBA achieved a greater yield when making the pure HIPS than in the previous study. No information is provided in the Fisher et al. study, however, on how much sorting and culling was done before materials were shipped to MBA—i.e., what percentage of the total plastics collected were shipped for further processing.

[22] The Macauley et al. study measured only benefits from reduced lead releases from incinerators, but the authors argue that these are the most important benefits from reduced disposal of CRTs because lead from incinerating CRTs far outweighs lead in leachate from landfilled CRTs. Although the authors do not measure any upstream life-cycle environmental benefits, they argue on the basis of their numerical results that those benefits would have to be several orders of magnitude greater than the direct end-of-life benefits to yield positive overall net benefits.

The net costs of recycling are lower when products are easier to disassemble, thus designing products for disassembly is important. (4) Finding ways to use secondary materials from electronics in the production of new electronics—i.e., closed-loop recycling—is critical. This happens to some extent with CRT glass, but more could be encouraged. Plastics face a high hurdle in this regard; at present, most plastics in electronics, if they are recycled at all, are recycled into much lower valued products such as lumber, outdoor furniture, and materials for roads.

Articulating these product design objectives is one thing; designing *feasible* policy instruments that achieve these objectives is quite another. Moreover, many policies that are intended to have one effect often turn out to have another. We return to this issue in greater detail after discussing maxim 2, the need for clarity in policy objectives.

5.3. Maxim 2: Clarifying policy objectives.

E-waste is a good example of how easy it can be to obscure the objectives of a proposed EPR policy. And obscuring the objectives can lead to costly policy choices.

I use the Macauley et al. (2001) study again for illustration. The authors assume in that study that the only measurable benefits from recycling CRTs come from reduced lead releases from incinerators. Because of this, they conclude that the most cost-effective policy option is one in which CRTs are banned from incinerators. Although this policy yields a low recycling rate compared to the other options the authors analyze, it is the most effective at reducing incinerator lead emissions and the least costly policy option on their list.[23] The highest recycling rate in the Macauley et al. study comes from a combined ban on all disposal, in landfills and incinerators, and a recycling subsidy of $10 per monitor. However, this is the most costly of the instruments they analyze. The authors do not address EPR-based instruments.

In discussions about policies directed at electronics waste, one often finds that participants have different views on policy objectives—i.e., different views on exactly which environmental problems and which electronics products should be the focus of policy. CRTs are an interesting case in point. Most of the e-waste legislation introduced at the state and federal level in the United States is focused on CRTs, or on computers more generally, and not broadly

[23] The authors do not address enforcement issues or the potential for illegal disposal. These problems could change their results.

on other electronics. If lead emissions are the primary environmental concern, as assumed by Macauley et al., this may be appropriate. On the other hand, if reducing waste volumes is the policy objective, it should be kept in mind that CRTs account for only about 12% of the weight of materials collected in electronics collection programs (U.S. EPA, 1998). Policy instruments thus need to have a broader target to cost-effectively reduce overall electronics waste disposal.

Even if lead emissions are the primary concern, policymakers need to be careful to devise policies that account for a number of market factors, not the least of which is the increasing market penetration of flat panel displays. Flat panels may make the lead disposal externality go away eventually on its own. Policymakers need to guard against the potential problem of devising a costly policy that quickly becomes, at best, moot and, at worst, stifling to innovation and growth.

Discussion in the NEPSI process, which is focusing on developing a voluntary electronics recycling system, have included disagreements over exactly which products should be targeted in the system. Some consensus seems to exist over CPUs, computer monitors, and some peripherals, but exactly which peripherals to include and whether other electronics products should be in the agreement are points of debate. There are also discussions, with no resolution as yet, on whether and to what extent the group will try to address DfE.

It is interesting that in the EU, the WEEE directive clearly targets overall waste reduction from a broad class of electrical and electronic products. The toxic constituents of such products are addressed, but in a separate directive, the Directive on Restriction on the Use of Certain Hazardous Substances (RoHS) in electrical and electronic equipment. This directive bans the use of lead, mercury, cadmium, hexavalent chromium, and some flame retardants beginning in 2008. It remains to be seen whether policy instruments in EU member countries will be developed that are cost-effective, but at least the EU seems to be abiding by maxim 2 in clarifying the environmental objectives and having multiple policies for multiple objectives.

5.4. Maxim 4: policy difficulties

At the same time that the complexity of products yields more potential for environmental product design, it also makes the job of the policymaker more difficult. Government officials, far removed from the product assembly line and the secondary materials processor, cannot be expected to know the ins and outs of designing a computer monitor, stereo, cell phone, or television. In many cases, it is difficult for producers themselves to know exactly what can be done to improve a product's recyclability. Clearly, advanced recycling of the plastics used in electronics, for example, is still in a fairly infant stage of development. Figuring out the optimal designs to address a multitude of safety, performance, quality, durability, *and* recycling concerns with those plastics is an exceedingly difficult job.

It is clear that any feasible and cost-effective policy that addresses DfE for electronics will have to count on market signals working, at least to some extent. So while poorly functioning recycling markets might present an argument for an EPR policy to spur DfE, as we explained above, this does not imply that government should try to devise policy that *replaces* markets. And almost any conceivable policy—even an individual take-back mandate with an accompanying recycling target—will still rely on market signals for the policy to work properly: signals from material suppliers to producers and vice-versa, signals from recyclers to producers, and signals from consumers to producers.

5.5. Returning to maxim 1 and policy recommendations for e-waste.

The above discussion might seem to suggest that feasible and cost-effective EPR policy alternatives for e-waste are limited, if not nonexistent. Although we need further study of incentive-based policies, such as tradable recycling credits, that have the potential to spur DfE, I continue to believe that a UCTS holds much promise. The product tax, assessed on a per-pound basis, would lead to lighter-weight, less material-intensive products. The recycling subsidy could be paid to collectors of e-waste, with payment made after the materials are accepted by a certified processor and designated as "clean" and ready for recycling. Again, it is crucial that the subsidy be paid on a per-pound basis. Lump-sum grants to cover the costs of municipal collection programs, as has been suggested in the United States, do not provide incentives, *at the margin*, for more recycling. It is essential that the recycling payments are greater, the greater the volume of materials recycled.

The UCTS, in combination with unit-based pricing of residential waste, should generate more e-waste diversion and has the possibility to cause some product design changes. These design changes will come about if, at some level of prices, recycling markets work properly—i.e., recyclers pay collectors prices for materials that vary with product recyclability and collectors, in turn, pass some portion of that price back to the original consumers who returned the products for recycling. Further study is warranted into how the government, through provision of information, can facilitate the functioning of these markets. Further study is also warranted into possible refinements of the UCTS instrument that might motivate more DfE—such as taxes that vary by material type—as well as the possibility of other kinds of financial incentives. In European and other countries that have espoused the take-back approach, more research is needed into the possibility of using tradable recycling credits in combination with take-back.

6. Concluding Remarks

Identifying, designing, and implementing cost-effective environmental policies is no small job. Accomplishing this goal when multiple environmental objectives are on the table, when it is vital that the policy spur DfE, and when some markets, such as recycling markets, function poorly is even more difficult. In this paper, I have tried to outline some maxims for policymakers: striving for cost-effectiveness and clearly defining policy objectives while doing so; using EPR-based instruments when either illegal disposal or poorly functioning recycling markets present themselves as problems; and recognizing and dealing with the fact that using policy to motivate DfE is always going to be difficult. With respect to the last maxim, it is important that policymakers not let the added cost of instituting a complex policy outweigh the added DfE benefits. Further research is needed on how to design incentive-based and low-cost policies that can generate more DfE.

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